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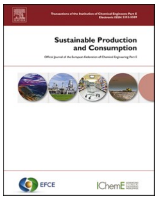

Towards a sustainable circular economy: Understanding the environmental credits and loads of reusing modular building components from a multi-use cycle perspective

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ABSTRACT

Properly designed modular construction offers the potential for easy disassembly, relocation, and reuse across multiple use cycles. However, the environmental benefits and burdens resulting from the reuse of modular components over these cycles are not well understood. The study aimed to assess the environmental credits and loads associated with reusing modular components over multiple use cycles. This aim was achieved through two approaches. Firstly, three dedicated life cycle assessment (LCA) allocation rules were adopted, namely, cut-off with Module D, the Product Environmental Footprint (PEF), and the Circular Footprint Formula (CFF), to evaluate the environmental impacts of production, reuse, repair, replacement, recycling, and disposal of a modular unit (including the steel frame, concrete slab, and steel connector) across different life cycle stages (Module A, Module C, and Module D) and use cycles (first, intermediate, and last). The PEF approach was determined to be the most suitable for interpreting the environmental credits and burdens associated with reuse. The study found that the reuse and recycling of the modular unit resulted in approximately 9007 ± 362 kg, 2925 ± 602 kg, and 8433 ± 544 kg of equivalent carbon dioxide emissions in the first, intermediate, and last use cycles, respectively. Secondly, a global sensitivity analysis was performed to assess how uncertain input parameters related to future use cycles (e.g., reuse rate, direct reusability rate, recyclability rate, and transport distance) influenced the LCA outcomes. The results revealed that it is beneficial to achieve a higher level of reusability (i.e., direct reusability) and recyclability for the steel frame to maximize the environmental advantages. The impact associated with a relatively lower level of reusability (i.e., repairable) and recyclability for subcomponents is considered environmentally acceptable. However, the lowest level of reusability of subcomponents (i.e., replaceable) should be avoided to minimize the impact associated with replacements. With a view to ensuring net environmental benefits from reuse, it is crucial to attain the desirable reusability level through developing proper design and deconstruction strategies for individual modular components.

Abbreviations

Acronyms

CFF	circular footprint formula
DfD	design for deconstruction
FPMF	fine particulate matter formation
GWP	global warning potential
LCA	life cycle assessment
LCI	life cycle inventory
OFHH	ozone formation, human health
OFTE	ozone formation, terrestrial ecosystems
PEF	product environmental footprint
TA	terrestrial acidification

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TET	terrestrial ecotoxicity
VRP	value retention process

Symbols used in equations

dl	distance of transport between site and landfill (km)
dp	distance of between site and offsite factories (km)
dw	distance of transport between site and warehouse (km)
dsort	distance of transport between site and sorting facilities (km)
drec	distance of transport between sorting facilities non-local recycling facilities
I_L	the impact of landfill
I_P	the impact of production of virgin material
I_{P*}	the impact of production of virgin material assumed to be substituted by recyclable materials
I_{rec}	the impact of the recycling process including collection, sorting, and treatment
I_{repair}	the impact of repairing
$I_{replace}$	the impact of replacing
I_{VRP}	the impact of the value retention processes
$I_{VRP,core}$	the impact of the value retention processes for the core component
$I_{VRP,sub}$	the impact of the value retention processes for the subcomponent
Q_p	the quality of primary material
Q_s	the quality of secondary material
R_1	rate of reuse
R_2	rate of reusability
R_c	rate of recyclability
R_d	rate of direct reusability
R_{recst}	recyclability rate of steel
R_{recon}	recyclability rate of concrete

1. Introduction

The construction industry traditionally follows an unsustainable linear economic model of “take, make, dispose” (Benachio et al., 2020). There has been a paradigm shift from this linear model to a circular model (Benachio et al., 2020), which aims to minimize waste, pollution, and greenhouse gas emissions while promoting the prolonged use of resources through extended product life cycles (Geissdoerfer et al., 2017). One of the circular economy principles is to optimize resource yields by reducing, reusing and recycling products, components, and materials, thereby creating a closed-loop system (Ellen MacArthur Foundation, 2013). Although the implementation of the circular economy has gained attention worldwide, it has predominately centered on recycling rather than reuse (Ghisellini et al., 2016) due to uncertainties associated with reusing materials once they have reached the end of their lives (Hossain et al., 2020). For instance, deconstructing conventional in-situ construction can cause damage to components and materials, making reuse impractical (Xia et al., 2020).

The Design for Deconstruction (DfD) is a practice that allows the ease of disassembly to salvage the materials and products for recycling or reuse through thoughtful planning and design (Rios et al., 2015). DfD enables connections and elements of a building to be dismantled and reused in another (or the same) building, making construction reversible (Dams et al., 2021). The ease of disassembly can be achieved by using bolted, screwed, and nailed connections instead of chemical or welding connections; while using prefabricated and/or modular components enables the separation of reusable items without causing significant damage (Guy and Ciarimboli, 2007; In Rios et al., 2015). The adoption of DfD supports the direct reuse of building components, requiring fewer additional carbon and energy inputs compared to recycling, and it becomes an essential approach to achieving a circular economy in the building sector (O’Grady et al., 2021).

The environmental benefits of DfD solutions compared to conventional structural design have widely been discussed. Eberhardt et al. (2019) assessed an office building designed for disassembly and found that DfD could save about 10 % embodied impact compared to the conventional scenario without reuse. Minunno et al. (2020) evaluated the life cycle environmental benefits of a modular building designed for disassembly and reuse. They found that reusing modular components could offset greenhouse gas emissions by 88 % compared to recycling. Roberts et al. (2023) examined the global warming potential of a DfD building, indicating that reusing the building’s panelized building fabric could avoid the production of the same component using virgin materials, resulting in a global warming potential reduction of 440 kgCO2-eq/m^2. Eckelman et al. (2018) found that DfD designs could have lower impacts than traditional designs if the DfD components were used at least once.

While the benefits of adopting DfD are promising, it remains unclear how many loads would be added for future reuse scenarios considering the impacts of value retention processes (VRPs) that enable reuse. VRPs are mechanisms that retain value in the economy through direct reuse, repair, refurbishment, remanufacturing, redistribution, and recycling (Nasr et al., 2018; In: Haupt and Hellweg, 2019). Different VRPs can offer varying degrees of process and resource-use intensity, resulting in varying environmental benefits and loads (Russell and Nasr, 2019). Direct reuse and repair require significantly fewer resources, while replacement and refurbishment can be resource-intensive (Russell and Nasr, 2019). It is of importance to understand the influence of different VRPs on the environmental performance of reusable building components when bringing those elements to a subsequent life cycle (van Stijn et al., 2021).

Prior studies have made their first attempts to advance methodologies relevant to evaluating the environmental impacts of reusable/recyclable building components/materials. De Wolf et al. (2020) compared the existing life cycle assessment (LCA) approaches to quantify the environmental impacts of reusable/recyclable building components. They concluded that the current LCA methods need to provide consistent assessment outcomes when evaluating building components that have or will have multiple life cycles. Eberhardt et al. (2020) confirmed that a linear degressive approach could address the benefits and loads of recyclable and reusable building components between multiple life cycles. Built on this study, van Stijn et al. (2021) further applied the linearly degressive approach to assess the impacts of the circular building components that are cascaded into different things. Obrecht et al. (2021) employed the product environmental footprint (PEF) to evaluate the environmental impacts of refurbishing building components; however, the application of the PEF in the context of reuse and recycling building materials and components is limited (e.g., Schaubroeck et al., 2022). In recent years, the circular footprint formula (CFF) proposed by the European Commission (2017) has emerged as an updated alternative to the PEF formula. While the PEF aims to distribute the environmental impacts of recycling/reuse equally between the product systems (or life cycles), the CFF shares these impacts based on market demand and supply factors (Schrijvers et al., 2021). Eberhardt et al. (2020) applied the CFF to assess the environmental impacts of reusing building components over multiple life cycles. They indicated that the CFF provided a strong incentive to prioritize design for disassembly in the first cycle, as less impact was allocated to that cycle. Given that the impact allocation methods can result in different assessment outcomes, it is critical to determine a proper one that achieves physical realism and ensures fair allocation of burdens and benefits between cycles, avoiding double-counting and inconsistency (Allacker et al., 2017). Before the assessment, this study would review various allocation methods (e.g., cut-off, avoided burden, product environmental footprint) to understand their suitability for assessing the impact of reusing building components across multiple use cycles (see Section 2).

Before assessing the environmental impact of reusable components over multiple-use cycles, it is also crucial to determine the reusability and recyclability potentials of building components to quantify the quantity of components that can be reused and recycled in future. Xia et al. (2020) introduced the degradation rate as a factor affecting the reusability of concrete components. Antwi-Afari et al. (2022, 2023) adopted a building systemic circularity indicator to forecast the percentages of recyclable materials. van Stijn et al. (2021) estimated the technical, functional, and economic lifespans of materials and components to determine repair, replacement, and remanufacturing rates in their LCA model. Despite these efforts, the reliability of predicting the reusability and recyclability of building elements remains debatable due to uncertainties in future use scenarios. For example, unpredictable damage to components during disassembly, lifting, and transportation may affect their reuse potential. Instead of predicting or determining uncertain factors, examining the influence of these uncertainties on the assessment outcomes may be more realistic and practical. One of the key methodological approaches of the LCA method in this study is to analyse the impact of uncertainties in future unknown cycles to better understand the variations in the LCA results.

This study aims to assess the environmental loads and credits of reusing modular components over multiple use cycles. To accomplish this goal, a review of existing LCA studies and standards was conducted, focusing on how various impact allocation rules address critical methodological issues related to building reuse. The environmental impacts of reusable modular components between multiple use cycles were then evaluated according to the selected allocation rules. Additionally, the uncertainty and sensitivity analyses were performed to determine how uncertain future reuse scenarios could affect the LCA outcomes. The assessment results were presented through a case study.

This research forms one of the pioneering studies that evaluate the environmental credits and loads of reusing modular components over multiple use cycles. It brings novelty in three key aspects. Firstly, the findings shed light on the sensitivity of environmental impacts associated with reuse, specifically related to the LCA allocation methods (e.g., cut-off, PEF, CFF). It improves the understanding of LCA practitioners regarding the influence of allocation methods on specific outcomes of building reuse scenarios. The results also provide insights into the physical realism, fairness, and practicality of the chosen allocation rules (Allacker et al., 2017), guiding LCA practitioners in choosing appropriate allocation approach for assessing building reuse. Secondly, the uncertainty and sensitivity analyses conducted in this study contribute to the existing body of knowledge by identifying significant factors that can enhance the environmental benefits or reduce the environmental burdens associated with reusing modular components. The influential factors provide valuable guidance to decision-makers, enabling them to understand how sustainable reuse can be attained through thoughtful design and deconstruction planning. Lastly, previous studies on the LCA of building reuse typically report deterministic results, even though scenarios analyses may yield a range of possible LCA outcomes, albeit sometimes limited. Compared to deterministic LCA results, this study contributes to a better understanding of the probabilistic life cycle environmental impacts throughout the multiple cycles of building reuse, particularly in the context of uncertain future scenarios.

2. Literature review

2.1. Life cycle assessment of building reuse

While recent attention has been given to the literature on LCA of building reuse, researchers and practitioners in the field have long focused on assessing the environmental impacts of recycling (De Wolf et al., 2020). When building components are properly designed, they can be directly reused in one or more subsequent building systems. However, unlike standardized manufactured products, building components are typically complex, unique, and unstandardized in different building systems. Thus, reusable building components have higher potential for being reused within the same structural system (i.e., a closed-loop system) than in a different system (i.e., an open-loop system).

The impact of the recycling strategy is usually assessed within an open-loop system, where material from one product system is recycled into a different one; it can also be evaluated within a closed-loop system, where material from one product system is recycled back into the same product system (Allacker et al., 2017). The life cycle assessment of material recycling has predominantly focused on an open-loop system, as most recycled materials will not return to the same building system (Xia et al., 2020). Recycled materials leave the system, and the future use of recycled materials occurs beyond the system boundary. In view of this, the system boundaries for life cycle assessment of reuse and recycling should be differentiated.

The modular building component combines non-replaceable (e.g., steel frame) and replaceable elements (e.g., concrete slab, connection). The non-replaceable components form the core, while replaceable components are the non-core elements that can be replaced without altering the core function. The lifespan of the component core determines whether the modular component can be reused or not (Tingley and Davison, 2012). While conducting an LCA of reusable modular building components, it is essential to differentiate between the core components and subcomponents. This distinction is crucial because they may be subject to various value retention processes (VRPs), such as repair and replacement, yielding different environmental impacts.

The allocation of impacts is a fundamental methodological issue for the life cycle assessment of building reuse. Allocation refers to “partitioning the input or output flows of a process or a product system between the product system under study and one or more other product systems” (ISO 14044, 2006). Because of missing consensus on allocation methods (Joensuu et al., 2022), allocation should be primarily avoided by (1) ‘subprocess modelling’ that divides the processes between the cycles and ‘cutting off’ the processes into two or more subprocesses, or (2) ‘system expansion’ that includes multiple cycles in the system boundary (ISO 14044, 2006). If the system expansion approach is applied, the impact of a reusable modular component across multiple use cycles can be expressed by the sum of the impacts of all the systems, thereby eliminating impact allocation. Finnveden et al. (2009) suggested that system expansion is suitable when LCA is employed to examine combinations of several life cycles. However, the assessment results become highly uncertain since future systems are still being determined. Subprocess modelling shares similar strengths and weaknesses with system expansion. Therefore, when dealing with the multiple life cycles of reusable components and the future systems or subprocesses are unknown, both system expansion and subprocess modelling are less preferable approaches.

The impact of reusable components can be assessed by appropriately distributing and allocating all the burdens and credits associated with their reuse across multiple life cycles (Eberhardt et al., 2019). This process involves allocating the loads resulting from reuse or recycling and the benefits derived from avoiding production and disposal burdens among the cycles. In the building sector, various allocation methods have been extensively discussed (De Wolf et al., 2020; Obrecht et al., 2021; Antwi-Afari et al., 2022, 2023; Eberhardt et al., 2020; Wang et al., 2023). For instance, De Wolf et al. (2020) compared six allocation methods and found that all of them indicated smaller global warming potential values during intermediate life cycles, highlighting the advantages of building reuse. However, they observed significant variations in values among the different allocation methods. Obrecht et al. (2021) examined five allocation methods and determined that the PEF method and the cut-off method with Module D were favorable compared to other methods. This is because they consider both the benefits and burdens of recycling and reuse, thereby promoting the circular use of materials. Table 1 provides an overview of the various allocation methods used to distribute the impacts of reuse or recycling across major life cycle stages/Modules, namely, Production (Module A), End-of-life (Module C), and Beyond the system boundary (Module D). However, there has yet to be a consensus on which allocation method is most suitable for assessing the impact of building reuse.

Table 1
Common impact allocation approaches for distributing the impact between life cycles.

Allocation method	Module A – production		Module C – EoL	Module D		References
	Virgin	Reused content	Disposal	Load	Credit	
Cut-off (100: 0)	$(1\text{-}R_1)I_P$	$R_1 \cdot I_{VRP}$	$(1\text{-}R_2)\ I_L$	0	0	Obrecht et al. (2021), BSI (2008)
Cut-off with Module D	$(1\text{-}R_1)I_P$	$R_1 \cdot I_{VRP}$	$(1\text{-}R_2)\ I_L$	$(R_2\text{-}\ R_1)\cdot I_{VRP}$	$-(R_2\text{-}\ R_1)\cdot\ I_P$	Obrecht et al. (2021)
Avoided burden (0:100)	I_P	0	$(1\text{-}R_2)\ I_L$	$R_2 \cdot I_{VRP}$	$-R_2 \cdot I_P$	Obrecht et al. (2021), BSI (2008)
Equal distribution (50:50)	$(1\text{-}R_1)I_P$	$R_1/2 \cdot I_{VRP}$	$(1\text{-}R_2)\ I_L$	$R_2/2 \cdot I_{VRP}$	0	Allacker et al. (2017), BSI (2008)
Product environmental footprint (PEF)	$(1\text{-}R_1/2)\ I_P$	$R_1/2 \cdot I_{VRP}$	$(1\text{-}R_1/2\text{-}\ R_2/2)I_L$	$R_2/2 \cdot I_{VRP}$	$-R_2/2 \cdot I_P \cdot\ (Q_s/Q_P)$	Allacker et al. (2017); Obrecht et al. (2021); European Commission (2013)
Simplified circular footprint formula (CFF)[a]	$(1\text{-}R_1)I_P$	$R_1/2 \cdot I_{VRP}$	$(1\text{-}R_2)\ I_L$	$R_2/2 \cdot I_{VRP}$	$-(R_2\text{-}\ R_1)/2\ \cdot I_P \cdot\ (Q_s/Q_P)$	Zampori et al. (2019); Schrijvers et al. (2021), European Commission (2017)
Equal distribution by the number of life cycles	I_P/N	$R_1/2 \cdot I_{VRP}$	I_L/N	$R_2/2 \cdot I_{VRP}$	0	Allacker et al. (2017), De Wolf et al. (2020)
Equal distribution by the length of the current cycle	$l_{use}/L \cdot I_P$	$R_1/2 \cdot I_{VRP}$	$l_{use}/L \cdot I_L$	$R_2/2 \cdot I_{VRP}$	0	Adapted from De Wolf et al. (2020), van Stijn et al. (2021)

Note: l_{use} is the length of the current cycle, L is the length of all use cycles, N is the number of life cycles, R_1 refers to the rate of recycled/reused content, R_2 refers to the rate of reusability/recyclability, I_P is the impact of production of virgin material, I_L is the impact of landfill, I_{VRP} is the impact of the value retention processes (e.g., transportation, repair, replacement, and recycling/reuse treatment) and it is assumed that the impact arising from the value retention processes equals to the impact arising from the production process of the recycled/reused material in a closed-loop system, Qp and Qs represent the quality of primary and secondary materials, respectively.

[a] Recovery of energy is not considered. The allocation factor is assumed as 0.5, indicating the equilibrium between supply and demand (Zampori et al., 2019). The same quality of the recycled material and the recyclable material at the point of substitution is assumed.

The cut-off approach, also known as the 100:0 approach, assigns the environmental impacts of the production phase for a product to its first life cycle. The intermediate use of the product only carries the environmental impact of reuse/recycling. However, the materials that are reused in future cycles do not bear any environmental loads from the primary production process (Frischknecht, 2010; Allacker et al., 2017), as the primary production is attributed to the first life cycle (van Gulck et al., 2022). According to the cut-off allocation approach, the burden of disposal is entirely allocated to the last life cycle, where the previous cycles do not bear any impact from disposal. Nevertheless, this approach only considers the reused or recycled content, while the environmental loads and benefits associated with reusable or recyclable content are disregarded (Allacker et al., 2017; Obrecht et al., 2021). To overcome this challenge, the cut off approach with Module D is developed to include possible benefits or loads due to the value retention processes (Obrecht et al., 2021).

The avoided burden or 0:100 approach allocates the credits and loads to the reused or recycled content, excluding the potential impact of reusable or recyclable content (Allacker et al., 2017). The 50:50 approach evenly divides the impact of reuse or recycling between the product (in the previous cycle) that produces the reusable or recyclable material and the product (in the subsequent cycle) that uses the recovered material. In this approach, the intermediate cycles do not bear the impacts of primary production or end-of-life disposal.

In contrast to the equal sharing of the reuse/recycling impact among life cycles, the linearly degressive approach assigns the largest portion of the initial production and disposal impact to the cycle where they occur (Allacker et al., 2017; van Stijn et al., 2021). Specifically, the share of the primary production impact allocated to the previous and subsequent cycles decreases linearly, while the share of disposal impact increases linearly across the cycles. Furthermore, the impact of reuse/recycling is evenly between successive cycles. In the linearly degressive approach, the allocation factor 1/N is applicable when the cycles are of equal length (Eberhardt et al., 2019). When the impacts are equally distributed among cycles but the cycles are not of equal length, the allocation factor $l_{use}/\sum l$ should be used (van Stijn et al., 2021). However, these approaches heavily rely on the prediction of the number of life cycles and the length of the current cycle.

The PEF method allocates the environmental impacts of primary production and disposal between the first and last cycles, while equally distributing the loads and credits associated with recycling or reuse between two successive cycles. Unlike the 50:50 approach, the PEF method considers the credit attributed to reuse or recycling in Module D. This method ensures that the impacts of primary production, reused/recycled content, disposal, and reuse or recycling, are adequately allocated. Furthermore, the PEF method distinguishes the impacts of reusability/recyclability (i.e., R_2 in Table 1) from those of reused/recycled content (i.e., R_1 in Table 1), allowing for a fair allocation of the environmental impacts. Moreover, the PEF eliminates the need to determine the number of cycles or the lifespan of a material/product. This practical feature of the PEF method is beneficial when future life cycles are uncertain (Allacker et al., 2017).

The CFF, developed by the European Commission (2017), serves as a replacement for the PEF method (Zampori et al., 2019; Schrijvers et al., 2021). The CFF enhances the cut-off with Module D formula by introducing an allocation factor that determines the distribution of burdens and credits of recycled materials between the supplier and user. Additionally, the CFF considers the quality degradation of recovered/recoverable material compared to primary virgin material (Eberhardt et al., 2020). By considering the market situation, the CFF provides insights into the demand and supply dynamics of recycled materials. It is particularly applicable to open-loop recycling scenarios where the recycled material is intended for use in cascade systems. The value of allocation factor represents the ratio of the supply of recycled materials to the demand. It is assumed that this value is 0.5 in this research because the modular unit in the case study has been reused in the subsequent life cycle, indicating that the supply of the reusable building components meets the demand of the reused components.

When considering the physical realism of the total impact of multiple life cycles, it is worth noting that the avoided burden approach tends to overestimates the total impact arising from the production process (see Table 1). On the other hand, all the other methods yield a total impact of approximately $\sum_1^N I_P + \sum_1^{N-1} I_{VPR} + \sum_1^N I_L$, which can represent the material flows over multiple life cycles, corresponding to mass balances and reflecting the physical realism (Allacker et al., 2017). This total impact can be viewed as the outcome of system expansion, which, however, requires the estimation of the life span of building components (van Stijn et al., 2021).

Although the two cut-off approaches, the three distribution methods, and the two product environmental footprint formulas achieve physical realism, they differ in how they allocate impacts across life cycle stages. For instance, the cut-off and distribution approaches do not consider credits at the end of a preceding life cycle. The major discrepancy among the cut-off with Module D, the PEF, and simplified CFF (when it is assumed that the allocation factor is 0.5) lies in their different approaches to distributing the production, value retention process, and disposal impacts across multiple life cycles. Specifically, the cut-off with Module D approach predominantly assigns primary production and disposal impacts to the last life cycle. The CFF method tends to primarily allocate the loads of reuse or recycling to the last life cycle. The PEF method evenly distributes production and disposal impacts between the first and last cycles, whereas it evenly allocates the reuse or recycling impacts between the two successive cycles. Furthermore, the cut-off with Module D does not consider the quality degradation of the material (Obrecht et al., 2021), while the PEF and CFF differentiate the quality of primary and secondary materials. Concerning the methodological issues discussed above, the cut-off with Module D, PEF, and simplified CFF formulas are preliminarily considered in this research because they can reflect physical realism and tend to allocate environmental loads and credits between life cycles in a fair manner.

2.2. Uncertainty in LCA

Assessing the environmental impacts of reusing modular components in future unknown cycles introduces considerable uncertainties in the multi-life cycle modelling process. The uncertainties associated with future unknown cycles pose a significant challenge in LCA (De Wolf et al., 2020; van Stijn et al., 2021). Previous research has attempted to address uncertainties, such as the reuse rate, using approaches like “what-if scenarios” (van Stijn et al., 2021) or “best-worst scenarios” (Rios et al., 2019). For example, van Stijn et al. (2021) analyzed the impact of varying the number of cycles, lifespans, and cycle number on the LCA outcomes of reusable and recyclable kitchen components. Antwi-Afari et al. (2023) conducted a sensitivity analysis of a modular steel slab, examining the effects of different scenarios for circular design (e.g., 0 %–100 % recyclability). Buyle et al. (2019) incorporated three refurbishment scenarios as part of a sensitivity analysis, exploring the influence of different frequencies of future refurbishments. However, it is challenging to predict the reusability or recyclability of building components in future unknown life cycles, especially in the case of unintentional damage. The limitations of the scenarios assessed may fail to encompass the complete range of potential impacts. Arbitrary assumptions made in such cases may lead to improper interpretation of the LCA outcomes. Instead of solely determining uncertain factors, this study intends to examine how these uncertainties affect the LCA results through comprehensive uncertainty and sensitivity analyses.

Uncertainty can arise from uncertain input data, such as temporal or geographical variations in life cycle inventory data (Guo and Murphy, 2012). Uncertainty analysis involves quantifying and propagating input uncertainties to output uncertainties (Igos et al., 2019). One common approach to studying the influence of uncertainties on output is through sensitivity analyses (Zhao et al., 2021). Sensitivity analysis techniques can be broadly categorized into local and global sensitivity analyses. Local sensitivity analysis involves varying each input parameter individually while keeping the others constant to examine the output variation. This approach has been widely used in literature due to its simplicity and low computational requirements (Hamilton et al., 2022). However, it assumes that the variation in one parameter is not associated with any change in the other parameters (Ferretti et al., 2016; Saltelli et al., 2010). For instance, Rios et al. (2019) conducted a local sensitivity analysis on variables including transportation distances, reuse rate, and number of reuses. They examined the influence of each variable on the assessment outcomes by holding the other two variables constant at their baseline value.

In contrast, a global sensitivity analysis evaluates the overall effects of each parameter on the model output by simultaneously varying all other model inputs. This approach is valuable for calculating the probabilistic LCA outcomes based on several uncertain inputs to determine the combined influence of each input on the output variance (Zhao et al., 2021). It clarifies how the key influential input variables affect output variance (Pannier et al., 2018). In this study, the global sensitivity analysis is considered more suitable than the local sensitivity analysis because it does not require holding other variables constant while assessing the sensitivity of a single variable. Instead, it allows for changes in other variables to be taken into account when determining the influence of an input variable on the output variance. This feature helps overcome the limitations of a local sensitivity analysis in the context of circular economy in construction. Concerning the limited scenarios assessed, the global sensitivity analysis can encompass a wide range of potential scenarios by introducing the probabilistic distributions of uncertain recycling rates, lifespans and pace of future refurbishment. The use of the global sensitivity analysis may also reduce time consumption compared to traditional local sensitivity analysis, especially when dealing with numerous and complex uncertain input variables.

3. Methods

3.1. Case study

A freestanding modular unit (Fig. 1) comprises a structural steel frame and a precast concrete slab. The modular units are connected using bolt and nut joints (Fig. 2). Such a “modularity” design achieves a DfD solution that allows the ease of assembly, disassembly, relocation, and reuse of the modular unit. The modular unit has dimensions of 12.2 m in length, 2.4 m in width, 1.9 m in height. The modular unit was manufactured using virgin materials without incorporating any recycled materials. The modular design therefore follows a “downstream reuse” concept, where virgin materials are used initially, but disassembly and reuse are prioritized after the building’s first life (De Wolf et al., 2020). Due to the lack of standardized modularity designs and connection systems in the market, reusing a specific modular unit in different modular systems is technically and economically challenging. Therefore, it is assumed that if the modular unit is functionally and technically reusable, it should be reused within the same building system in subsequent use cycle(s). If it is unsuitable for reuse, reaching the end of their designed lifespan, the modular unit should be demolished for recycling and disposal.

3.2. Goal and scope definition

The life cycle assessment was conducted to evaluate the environmental impact of reusable modular components and to determine the benefits and burdens arising from their multiple use cycles. This study defines a use cycle as the period starting from the production and/or assembly of modular components and ending with disassembly and/or disposal. Through the use cycles, the inherent properties of the modular components are unchanged, and the materials are intended for direct reuse or use after repair or replacement within the same system. The assessment focused on the modular unit’s core component (steel frame) and subcomponents (concrete slab and bolt-nut connector). The functional unit chosen in this LCA is the modular unit with a reference service period (i.e., the designed lifetime) of 50 years.

The LCA is conducted to evaluate the environmental burdens and credits associated with the reuse of these modular components across multiple cycles. Additionally, the impact of recycling non-reusable components is also evaluated. The life cycle stages considered in this study include the production of virgin materials (Module A), landfill disposal (Module C), and loads and benefits resulting from reuse and recycling (Module D). In Module D, the impacts of the value retention processes for reusable components and recyclable materials are considered, while the impact of the use of recycled materials is beyond

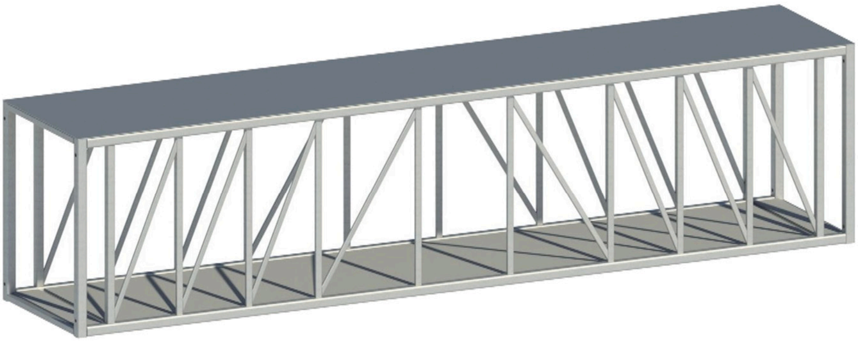

Fig. 1. Schematic diagrams for a typical modular component.

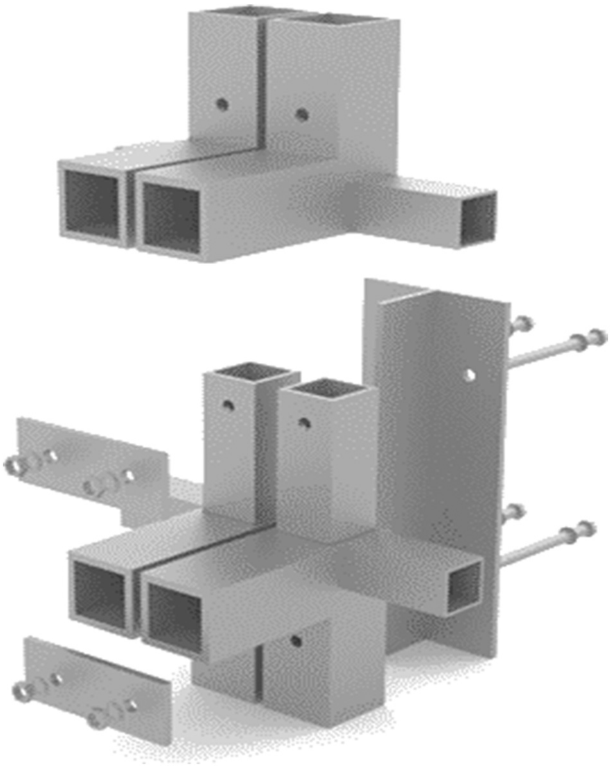

Fig. 2. Connection system of the modular unit.

the system boundary as it is assumed that those recycled materials would not be reused in the same building system. The assembly and disassembly processes and the operation phase (Module B) are excluded from this analysis. The system boundary is depicted in Fig. 3.

The Global warming potential (GWP, in kg co2-eq) is an indicator to assess equivalent carbon dioxide emissions and evaluate the carbon impact of a product, which includes all emitting gases: carbon dioxide (CO2), methane (CH4), chlorofluorocarbons (CFC), nitrous oxide (N2O), etc. The Ozone formation, Human health (OFHH, in kg NOx eq), Fine particulate matter formation (FPMF, in kg PM2.5 to air eq), Ozone formation, Terrestrial ecosystems (OFTE, in kg NOx eq), Terrestrial acidification (TA, in kg SO2 to air eq), and Terrestrial ecotoxicity (TET, in kg 1,4-DCB to industrial soil eq) are also viewed as the top priority impact categories for building materials (Feng et al., 2023) and thus, these six impact categories are assessed in this research.

3.3. *Life cycle inventory*

The life cycle inventory (LCI) analysis involves converting the processes associated with multiple use cycles into input and output materials. Each use cycle specifies the input as virgin and/or reused materials. It also defines which materials are reusable in the intermediate use cycle, which are disposed of at the end of the cycle, and which are recyclable. This step enables tracking material flows across the successive use cycles of the modular component.

To gather the LCI data, the input and output material flows within the system boundary were surveyed with site engineers involved in the disassembly and relocation process. To be more specific, 2D design drawings were obtained from the project team. Material quantities of the steel frame, concrete slab, and bolt-nut connectors are shown in Table 2.

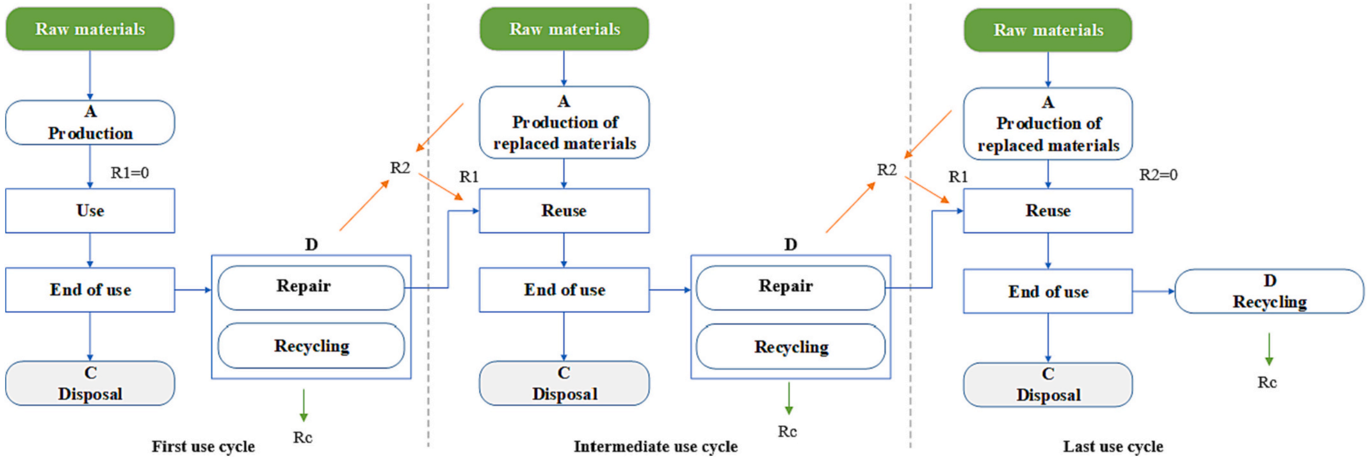

Fig. 3. System boundary of reusing the modular unit with multiple use cycles
Note: R_1 refers to the rate of reuse, R_2 refers to the rate of reusability.

Table 2

Quantities of major materials of the modular unit.

Component	Material	Weight (ton)	Volume (m^3)	Area (m^2)
Steel structure	Steel (S355)	6.59	0.84	207.24
Concrete slab	Concrete (C30)	4.83	1.93	40.92
Connectors (steel tube and plate, bolt & nut)	Steel	0.88	0.11	17.65

Table 3

Impact allocation between life cycle stages using the PEF formula.

Life cycle stage	Module A – production		Module C – EoL	Module D	
Content	Virgin	Reused content	Disposal	Load due to reuse and recycling	Credit due to reuse and recycling
Expression	$(1-R_1/2)\cdot I_P$	$R_1/2\cdot I_{VRP}$	$(1-R_1/2-R_2/2-R_c/2\cdot(1-R_2))\,I_L$	$R_2/2\cdot I_{VRP}+R_c/2\cdot(1-R_2)\cdot I_{rec}$	$-R_2/2\cdot I_P\ -R_c/2\,(1-R_2)\cdot\frac{Q_s}{Q_p}I_{P*}$
Steel frame 1st cycle	I_P	0	$I_L/2$	$I_{VRP}/2$	$-I_P/2$
Steel frame Intermediate cycle	$I_P/2$	$I_{VRP}/2$	0	$I_{VRP}/2$	$-I_P/2$
Steel frame last cycle	$I_P/2$	$I_{VRP}/2$	$(1-R_c)/2\cdot I_L$	$R_c/2\cdot I_{rec}$	$-R_c/2\cdot\frac{Q_s}{Q_p}I_{P*}$
Subcomponent 1st cycle	I_P	0	$(1-R_2/2-R_c\cdot(1-R_2)/2)\,I_L$	$R_2/2\cdot I_{VRP}+R_c/2\cdot(1-R_2)\cdot I_{rec}$	$-R_2/2\cdot I_P\ -R_c/2\,(1-R_2)\cdot\frac{Q_s}{Q_p}I_{P*}$
Subcomponent Intermediate cycle	$(1-R_1/2)\cdot I_P$	$R_1/2\cdot I_{VRP}$	$(1-R_1/2-R_2/2-R_c/2\cdot(1-R_2))\,I_L$	$R_2/2\cdot I_{VRP}+R_c/2\cdot(1-R_2)\cdot I_{rec}$	$-R_2/2\cdot I_P\ -R_c/2\,(1-R_2)\cdot\frac{Q_s}{Q_p}I_{P*}$
Subcomponent last cycle	$(1-R_1/2)\cdot I_P$	$R_1/2\cdot I_{VRP}$	$(1-R_1/2-R_c/2)\,I_L$	$R_c/2\cdot I_{rec}$	$-R_c/2\cdot\frac{Q_s}{Q_p}I_{P*}$

Note: R_1 refers to the reuse rate, R_2 refers to the reusability rate, R_c refers to recyclability rate, I_P is the impact of production of virgin material, I_L is the impact of landfill, I_{P*} is the impact of production of virgin material assumed to be substituted by recyclable materials, I_{VRP} is the impact of the value retention processes, I_{rec} is the impact of the recycling process including collection, sorting, and treatment. The impact of transport is included in all the processes. Q_p is the quality of primary material, Q_s is the quality of secondary material.

Coefficients of the six impact categories corresponding to specific processes and activities were collected from the Ecoinvent 3.9.1 cut-off database and Industry Data 2.0 database (Supplementary Information SI1).

The VRPs for the modular component are classified into (direct) reuse, repair, replacement, and recycling. While the structural steel frame can achieve a high rate of reuse, repairs or touch-ups (e.g., recoating, rectification, and polishing) may be necessary to maintain the building components in good condition after a period of time of use. Non-reusable concrete slabs and bolt-nut connectors can be replaced owing to the advantages of modularity design. The replacement process involves consuming virgin materials to replace the obsolete components and includes recycling and/or disposal treatment for the replaced obsolete objects. When modular components are no longer suitable for reuse, they are treated as conventional demolition waste, undergoing the sorting, recycling and landfilling processes.

To account for uncertainties related to the direct reusability rate, replacement rate, repair rate, recyclability rate and disposal rate in future use cycles, a probability distribution was formulated for each input parameter to represent the uncertainties associated with the variable (see details in Section 3.5). The case building is situated in Hong Kong. Potential local factories for the value retention processes can be found in CIC (2021). Potential local sorting facilities can be referred to the Civil Engineering and Development Department of Hong Kong S.A.R. (2023). The locations of landfills in Hong Kong can be obtained from the Environmental Protection Department of Hong Kong S.A.R. (2023). Non-local production factories and recycling facilities are assumed in Guangzhou District, Mainland, considering the principle of proximity. Given that the exact locations of the facilities above are unknown, the uncertainty in transport distances is considered (see details in Section 3.5).

3.4. Life cycle impact analysis

The impact analysis of reuse assumes a closed-loop system, where the emissions generated from the production of modular components using virgin materials are equivalent to the emissions associated with acquiring virgin modular components which would be substituted by reusable ones. The non-reusable modular component may have the potential for recycling. As explained in Section 2.1, the reusable modular component can be reused within the same building system in the next life cycle. In contrast, recyclable materials are not reused in the same system. Using the PEF approach, half of the recycling burdens and benefits are allocated to the existing building system, while the other half related to the cascade product system is not considered. In other words, recycled content is disregarded within the system boundary. The impact analysis of the three life cycle stages according to the PEF approach is expressed as Eqs. (1)–(3). Table 3 provides the details of the impact analysis for each life cycle stage of each use cycle for each modular element.

Module A – Production

$$I_A = \left(1-\frac{R_1}{2}\right)I_P + \frac{R_1}{2}I_{VRP} \tag{1}$$

Module C – End of life

$$I_C = \left(1-\frac{R_1}{2}-\frac{R_2}{2}-\frac{R_c}{2}(1-R_2)\right)I_L \tag{2}$$

Module D – Load and credit associated with reuse and recycling

$$I_D = \frac{R_2}{2}(I_{VRP}-I_P) + \frac{R_c}{2}(1-R_2)\left(I_{rec}-\frac{Q_s}{Q_p}I_{P*}\right) \tag{3}$$

where R_1 refers to the rate of reuse (%), R_2 refers to the rate of reusability (%), R_c refers to the rate of recyclability (%), I_A, I_C, and I_D is the impact of production, disposal, and the impact of reuse or recycling, respectively, I_P is the impact of production of virgin material, I_L is the impact of landfill, I_{P*} is the impact of production of virgin material assumed to be substituted by recyclable materials, I_{VRP} is the impact of the value retention processes (including repair and replacement), I_{rec} is the impact of the recycling process including collection, sorting, and treatment. The impact of transport is included in all the processes. Q_p is the quality of primary material, Q_s is the quality of secondary material.

The modular component is produced using virgin materials in the first use cycle, where no recycled/reused content is used ($R_1 = 0$ %). The reusable component is then reused in the subsequent use cycle, represented by R_2. In the last cycle, the modular component is no longer reusable as it has reached its lifespan, resulting in $R_2 = 0$ %.

Q_p and Q_s represent the quality of primary and secondary materials, respectively. The ratio Q_s/Q_p, known as the quality correction ratio, reflects the disparity in quality between the secondary and the primary materials. The ratio can be determined based on various factors, including the physical properties or the economic value difference between the materials (Allacker et al., 2017). Structural steel retains all of its metallurgical properties when recycled, meaning that the quality of recycled steel is equivalent to that of virgin steel (AISC, 2017). As for concrete, it can be downcycled as recycled concrete aggregate (RCA). Studies have shown that RCA costs \$2.40 per tonne, while natural aggregate costs \$4.70 per tonne (Frondistou-Yannas, 1981). Concerning the economic value difference, we assume a quality correction ratio of 50 % for recycled concrete.

As outlined in Section 2.1, it is necessary to conduct separate impact analyses for the core component (steel frame) and subcomponent (concrete slab, connector). In the case of the steel frame, a reusable steel frame implies that the reusability rate R_2 would equal 100 %. On the other hand, if the steel frame is not reusable, R_2 should be set as 0 %. In the intermediate cycles, both $R_1 = R_2$ would be assigned a value of 100 %. The reusability rate R_2 encompasses the direct reusability rate R_d and the rate of repair (1-R_d). It is worth noting that the welded steel frame, being the core component, is not replaceable during the preceding cycles. The impact of the value retention processes $I_{VRP,core}$ for the steel frame is assessed by the impact of repair I_{repair} (Eq. (4)).

$$I_{VRP,core} = (1 - R_d) I_{repair} \tag{4}$$

The subcomponents, including the concrete slab and steel connector, offer the option of either repair or replacement as part of the value retention processes. Consequently, the reusability rate R_2 of subcomponents is determined by combining the direct reusability (R_d) and repair rates (R_2-R_d). On the other hand, the replacement rate can be expressed as the rate of non-reusable materials (1-R_2). The non-reusable materials have the potential for recycling, with a recyclability rate R_c. Consequently, the impact of VRPs for subcomponents $I_{VRP,sub}$ is the sum of the impacts of repair I_{repair} and replacement $I_{replace}$, according to Eq. (5).

$$I_{VRP,sub} = (R_2 - R_d) I_{repair} + (1 - R_2) I_{replace} \tag{5}$$

Following the same logic of applying the PEF formula, details of the impact analysis using the simplified CFF and the cut-off with Module D methods are provided in Supplementary Information SI2.

3.5. *Interpretation of results*

The Global Sensitivity Analysis can be organized into four steps, as illustrated in Fig. 4. The first step chooses the probability density function (PDF) for each input parameter. The second step involves conducting a Monte Carlo simulation, which generates random values from the specified PDF for each input parameter (Table 4). In the third step, the impact analysis is performed to compute the output of the LCA model, specifying the environmental impacts of reusing modular components. Finally, sensitivity indices are estimated for each parameter using the Sobol's sensitivity analysis, indicating the key influential parameters affecting the LCA outcomes.

The probability density function (PDF) selection for each input parameter was based on various sources and assumptions. For instance, for input parameters like the rate of direct reusability, we assigned a uniform distribution defined by the range between the minimum and maximum values (Pannier et al., 2018). Considering that the location of the building site within Hong Kong for future cycles has not been determined, we opted to select the approximate center of Hong Kong as the chosen job site location. Similarly, we selected the approximate centers of cities in Guangdong province as the non-local construction and demolition (C&D) recycling facilities. The average distances from the site to corresponding facilities, such as factories, landfill facilities, warehouses, local separators, and non-local C&D recycling facilities, were assigned using a normal distribution (Pannier et al., 2018). In the case of the reused rate of the steel frame, we used a categorical distribution as it only defines two possible values: 1 for reusable and 0 for non-reusable (Pannier et al., 2018).

After identifying the probability distributions of the input parameters, we applied the Monte Carlo technique to generate a set of random values following these distributions. Monte Carlo simulation is the most commonly recommended for uncertainty analysis (Guo and Murphy, 2012). Specifically, we conducted a Monte Carlo simulation to generate 5000 random values from the specified PDF for each input parameter (Groen et al., 2014; Saltelli et al., 2008). The Monte Carlo analysis involves randomly sampling the probability distribution of each uncertain parameter; the probabilistic results are then computed based on the random input. This process allows for the propagation of uncertainties from the input parameters through the LCA model, resulting in a sample of model outputs. By executing this procedure, we constructed the probability distribution of the input parameters and computed the probability distribution representing the model results. The Monte Carlo simulation and uncertainty propagation were performed using Matlab.

A subsequent global sensitivity analysis was performed using Sobol's

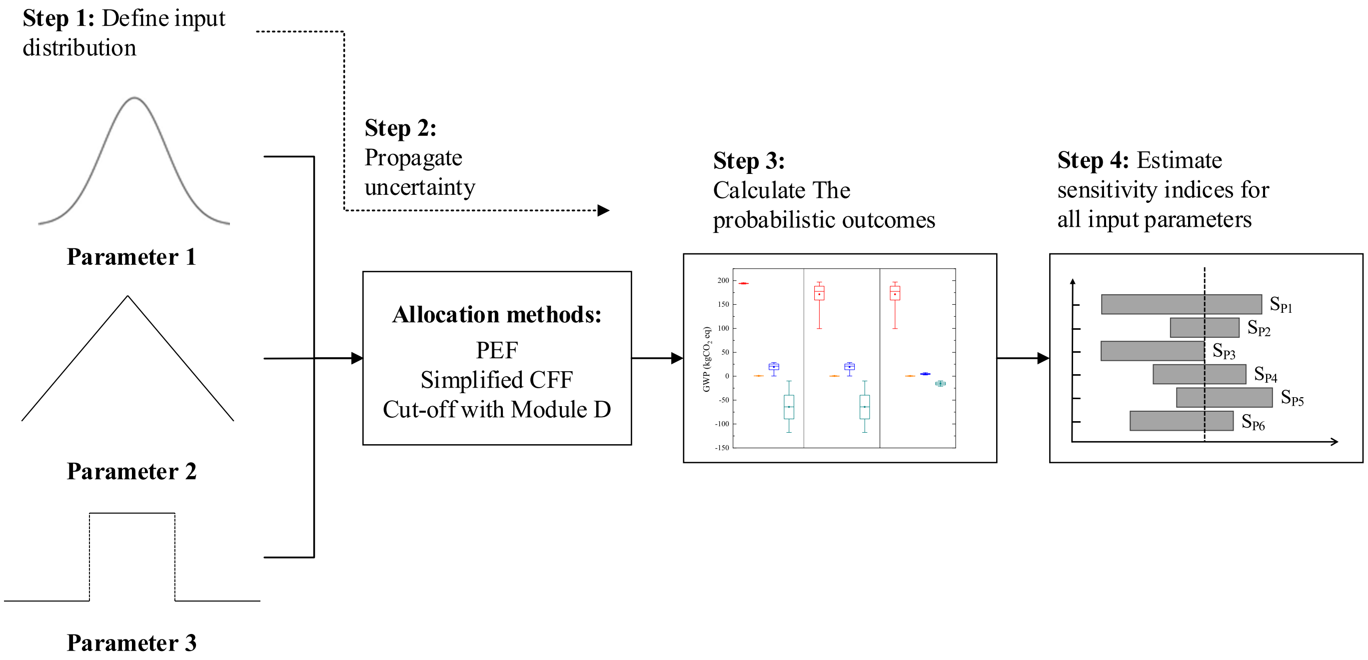

Fig. 4. Methodological framework of the global sensitivity analysis, adapted from Pannier et al. (2018) and Zhao et al. (2021).

Table 4

Assumed probability distribution of input parameters.

Variable	Description	Distribution	Mean (SD)	Min	Max	Data sources
n.a.	Rate of reused steel frame	Categorical	n.a.	0	1	Assumption
n.a.	Rate of reusable steel frame	Categorical	n.a.	0	1	Assumption
R_1	Rate of reused subcomponent (%)	Uniform	n.a.	0	1	Assumption
R_2	Rate of reusable subcomponent (%)	Uniform	n.a.	0	1	Assumption
R_d	Rate of direct reusability (%)	Uniform	n.a.	0	1	Assumption
R_{recst}	Recyclability rate of steel (%)	Uniform	n.a.	0	1	Assumption
R_{recon}	Recyclability rate of concrete (%)	Uniform	n.a.	0	1	Assumption
dl	Distance of transport between site and landfill (km)	Normal	33.6 (11)	21	41	Estimation of landfill facilities' distance based on Google map
dp	Distance of between site and factories (km)	Normal	168.6 (24.3)	141	187	Estimation of distance between site and non-local factories based on Google map
dw	Distance of transport between site and warehouse (km)	Normal	21.6 (11.7)	3.6	37	Estimation of potential warehouses' (CIC, 2021) distance based on Google map
dsort	Distance of transport between site and sorting facilities (km)	Normal	19.7 (24.2)	2.6	36.8	Local recyclers for Construction and Demolition (C&D) Materials
drec	Distance of transport between sorting facilities and non-local recycling facilities (km)	Normal	121.05 (63.7)	43	259	Estimation of distance between Hong Kong and cities in Guangdong province based on Google map

indices to determine which parameters are responsible for most of the variability of the LCA results (Sobol, 2001). The Sobol method provides two primary measures of sensitivity: the first-order effect (Si) and the total order effect (St). The first-order effect (Si) reflects the main effect of a parameter, indicating how much the output variance can be induced by fixing that specific parameter alone. Si represents the contribution of each input variable to the output variance when considered independently of all other variables. In contrast, the total order effect (St) considers the main parameter effect and the interaction effects with other parameters. Specifically, St represents the contribution of each input variable to the output variance when all possible combinations of the variable with other variables are considered. The Salib library within the Python 3.11 environment was utilized to compute the Sobol's indices.

4. Results

4.1. Probabilistic life cycle outcomes of modular components across multiple life cycles

The probabilistic GWP of the modular unit, including the steel frame, the concrete slab and connector, during each use cycle and each life cycle stage are shown in Fig. 5. Supplementary Information SI3 provides the results of the five other impact categories (OFHH, FPMF, OFTE, TA and TET). The LCA outcomes using the PEF, CFF, and cut-off with Module D allocation rules exhibit both similarities and differences. In both PEF and CFF allocation methods, reusing and recycling the modular unit result in approximately 9000 kg, 2900 kg, and 8400 kg of equivalent carbon dioxide emissions in the first, intermediate, and last use cycles, respectively. This finding suggests that the intermediate use cycle generates considerably fewer net impacts than the first and last cycles, indicating substantial environmental credits associated with reuse (De Wolf et al., 2020). According to the cut-off with Module D method, the emissions in the first, intermediate, and last use cycles are approximately 2405 ± 730 kg, 2736 ± 727 kg, and 13,750 ± 933 kg of equivalent carbon dioxide, respectively. The result obtained through the cut-off with Module D approach shows that the environmental burdens of reuse are primarily allocated in the last cycle, while the environmental benefits of reuse tend to offset the impact of primary production in the first cycle. Similarly, the CFF approach allocates the environmental credits associated with reuse in the first cycle but assigns the loads to the last cycle. On the other hand, the PEF tends to evenly distribute the environmental loads and credits associated with reuse between two successive cycles. The total impact values indicate that the PEF and CFF tend to distribute the environmental impacts of primary production and disposal between the first and last life cycles, while the cut-off with Module D allocates them primarily to the last cycle.

Regarding all allocation rules, the environmental credits associated with reuse are assigned to the first use cycle. According to the PEF and CFF approaches, when it comes to the core modular component (i.e., steel frame), the environmental credits can offset half of primary production (Fig. 6, Table SI3.2). With the cut-off with Module D method, these credits tend to offset all the primary production impacts. Compared to reusing the core component, the reuse of non-core modular components, such as concrete slab and connector, results in lower net environmental credits that offset less production impacts (Fig. 6, Tables SI3.3–S3.4). This is because the value retention processes, such as repair and replacement, can contribute to marked environmental loads. Despite this, in the first cycle, all allocation methods demonstrate that the environmental benefits derived from reuse outweigh the burdens associated with value retention processes.

The use of the three allocation methods yields similar net impacts in the intermediate use cycle. The PEF suggests that the environmental credits associated with reusable components can avoid the production of new components. In contrast, the CFF and cut-off with Module D distribute minimal benefits of reuse in the intermediate cycle (Fig. 5, Table SI3.1). Specifically, the CFF and cut-off with Module D methods indicate zero environmental credit associated with reusing the steel frame in the Module D-benefit stage (Fig. 6, Table SI3.2). Additionally, there is a significant variation in the environmental credits of reusing and recycling subcomponents in the Module D-benefit stage (Fig. 6, Tables SI3.3–SI3.4). Negative impacts are observed when the rate of reusability exceeds the rate of reuse. In contrast, positive environmental impacts are exhibited if the rate of reusability is lower than the rate of reuse, implying that decreased material efficiency from higher reused content to lower reusable content can result in environmental loads. Despite this, the CFF and cut-off with Module D approaches may face limitations in distinguishing environmental credits and loads attributed to reuse, as the environmental credits are treated as positive in certain scenarios. This situation becomes particularly pronounced in the last cycle when the rate of reusability reaches zero.

In the final use cycle, the PEF approach does not assign any environmental loads or benefits related to reuse since the modular components are no longer reusable. It only allocates half of the recycling benefits to the Module D stage, while the other half is allocated to the system beyond. Similar to the PEF approach, the CFF allocates half of the environmental loads and credits from recycling to the last cycle. However, the cut-off with Module D method assigns the full environmental loads and credits from recycling to the last cycle. According to the CFF rule, the Module D-benefit stage demonstrates positive impacts when the rate of reusability reaches zero (Fig. 5, Table SI3.1). It means that the CFF assigns the environmental burdens associated with reuse to the last cycle, considering that a zero reusability rate signifies the reduced material efficiency. It appears that the cut-off with Module D approach fails to differentiate and misinterprets the environmental loads and credits

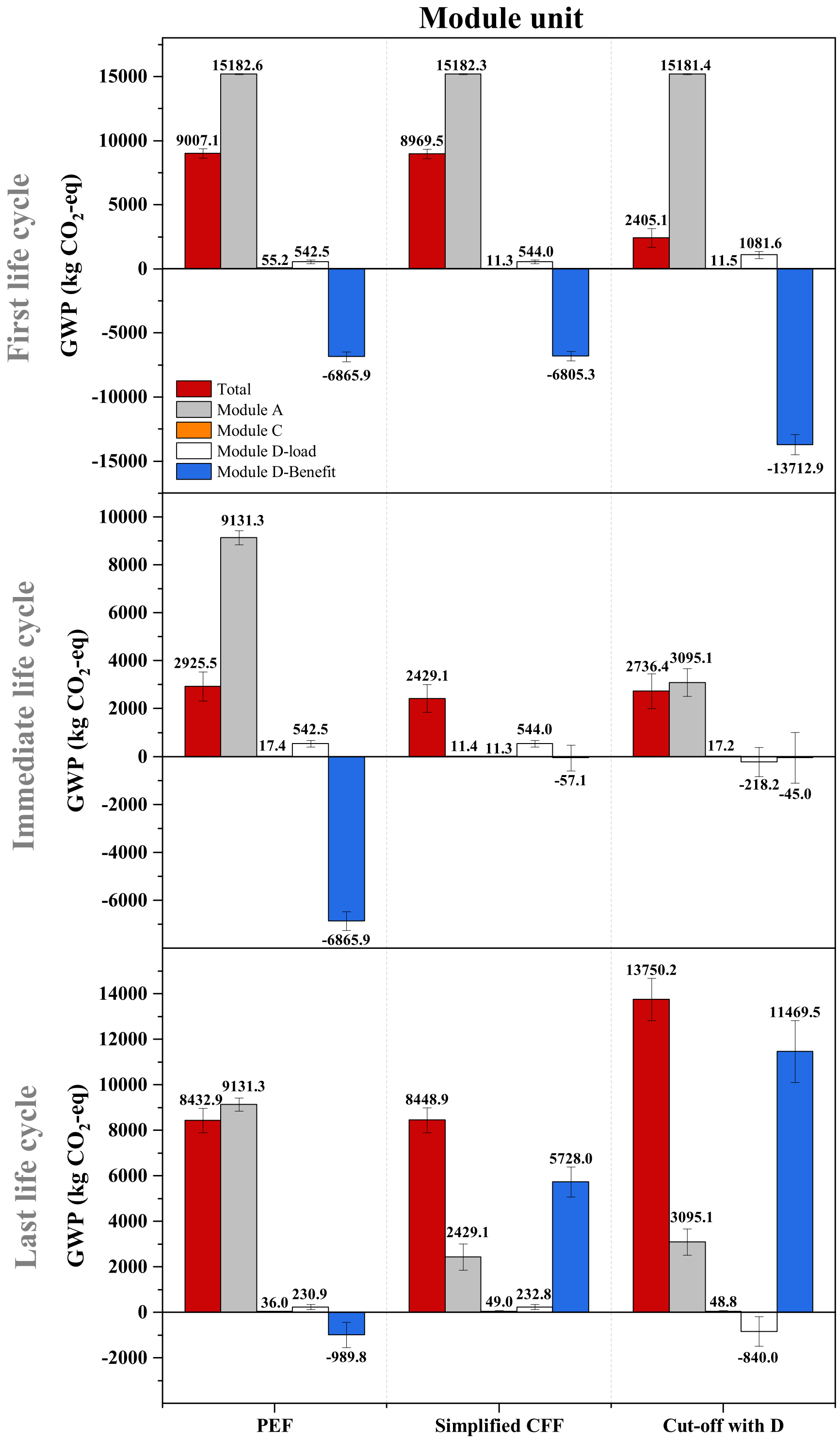

Fig. 5. Probabilistic life cycle CO2-eq emissions of the modular unit.

associated with reuse in the last cycle. Specifically, the impact of Module D-load tends to be negative because the environmental loads resulting from the value retention processes are treated as credits. On the contrary, the impact of Module D-benefit remains positive because the remaining half of the primary production impacts cannot be offset by the benefits of reused content.

The above results suggest that the PEF is more suitable than the CFF and cut-off with Module D allocation rules for assessing the environmental loads and credits associated with reusing building components because of the following reasons. The PEF clearly differentiates and properly interprets the positive and negative environmental impacts in each life cycle stage and each use cycle, providing a better understanding of the environmental consequences of reuse. The PEF acknowledges the reuse of building components in the preceding cycles, thereby promoting the circular use of these components. The PEF method also represents the environmental loads and credits resulting from recycling without considering reuse in the final use cycle. This interpretation aligns with physical realism since there are no more

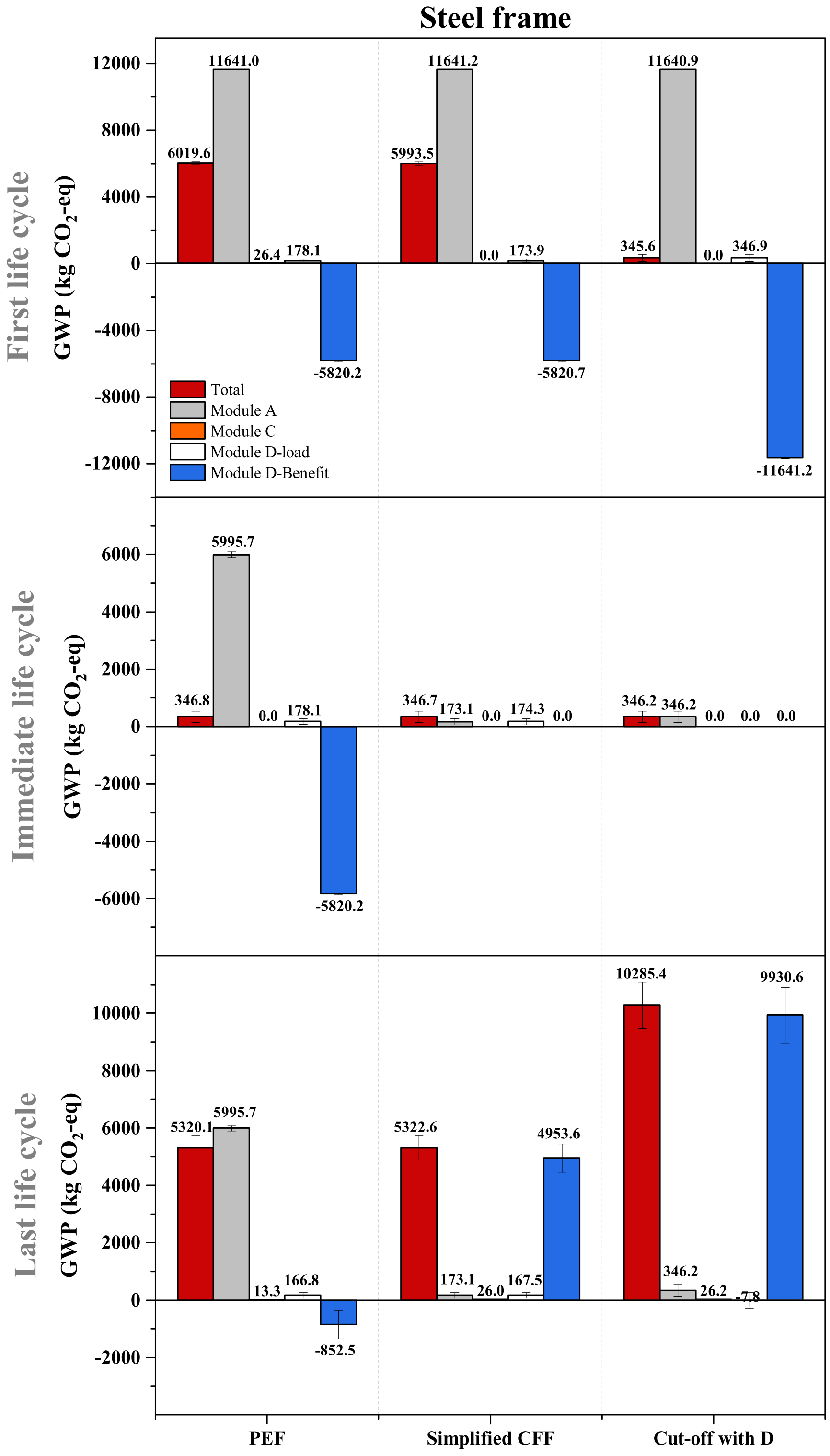

Fig. 6. The probabilistic life cycle CO2-eq emissions of individual modular elements in each use cycle and each life cycle stage.

reusable components available, and therefore, no further impacts are generated due to reuse. Considering these issues, the PEF allocation rule is selected for further sensitivity analysis as it is deemed more appropriate for assessing the environmental impacts of reusing building components. To this end, the following global sensitivity analysis was performed based on the PEF method.

4.2. Factors affecting the life cycle environmental impacts of reusing modular components

Supplementary information SI4 provides the results of the Sobol's indices for all impact categories. Table 5 provides a summary of the sensitivity factors for each individual modular component across different impact categories. The direct reusability rate (R_d), recyclability

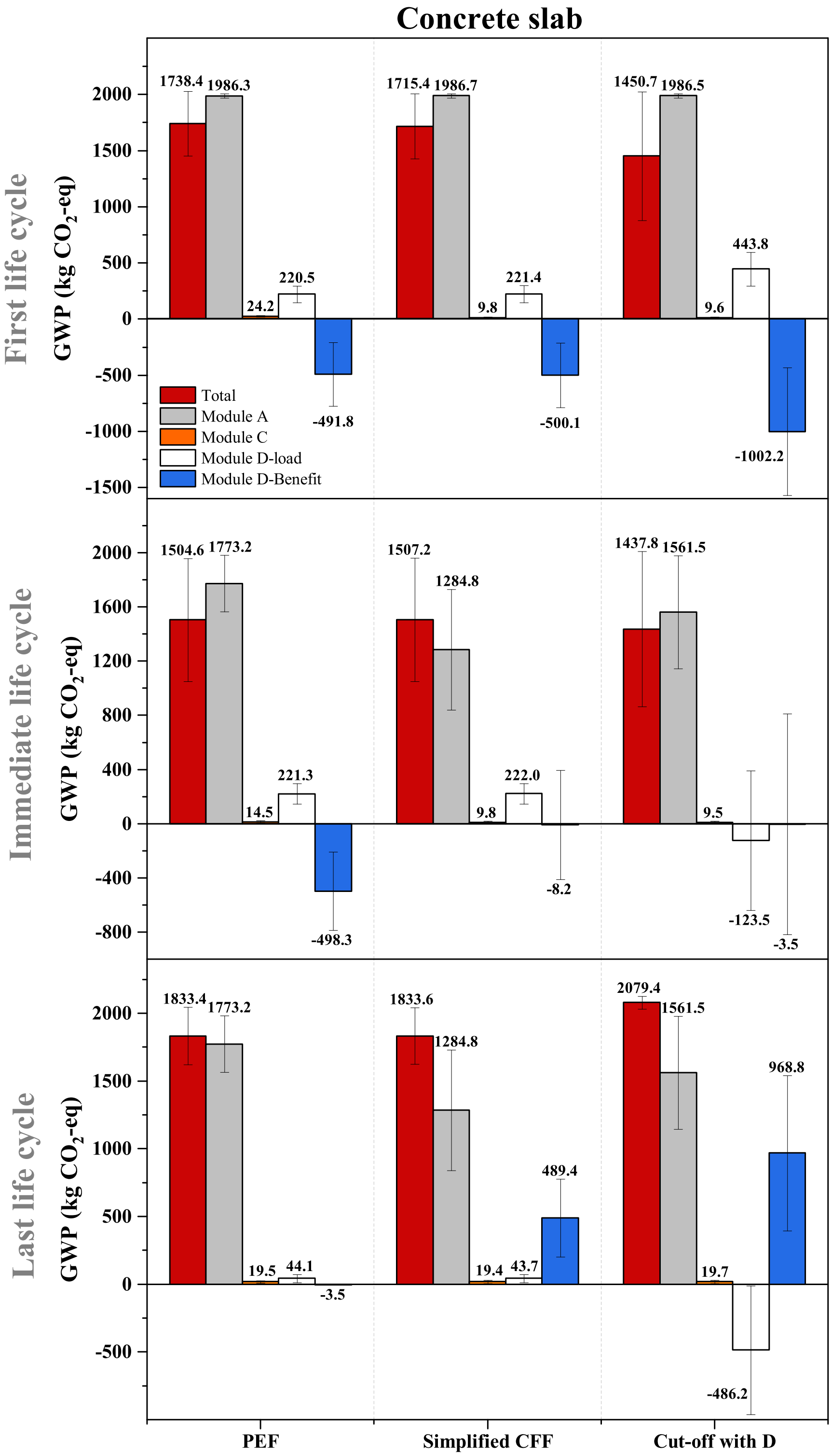

Fig. 6. (*continued*).

rate (R_c), and reusability rate (R_2) are identified as the most sensitive factors affecting the LCA outcomes. Among these factors, the direct reusability rate of the steel frame is the most influential factor across all impact categories. A lower level of direct reusability of the steel frame, which corresponds to a higher proportion of repairs, can significantly contribute to the environmental impact. The reusability rate of subcomponents emerges as the most sensitive factor influencing the LCA results across all impact categories. This finding indicates that a lower level of reusability of subcomponents, accompanied by a higher proportion of replacements, has a substantial impact on the environmental outcomes. However, variations in the proportion of subcomponents repairs do not significantly change the environmental impacts. These results suggest that the impact of the core modular component, such as the steel frame, is more sensitive to repairs in terms of its substantial influence on the environmental outcome. On the other hand, the environmental outcomes of subcomponents are more sensitive to the impact of replacements. Furthermore, changing the recyclability rates of subcomponents does not lead to significant alterations in the

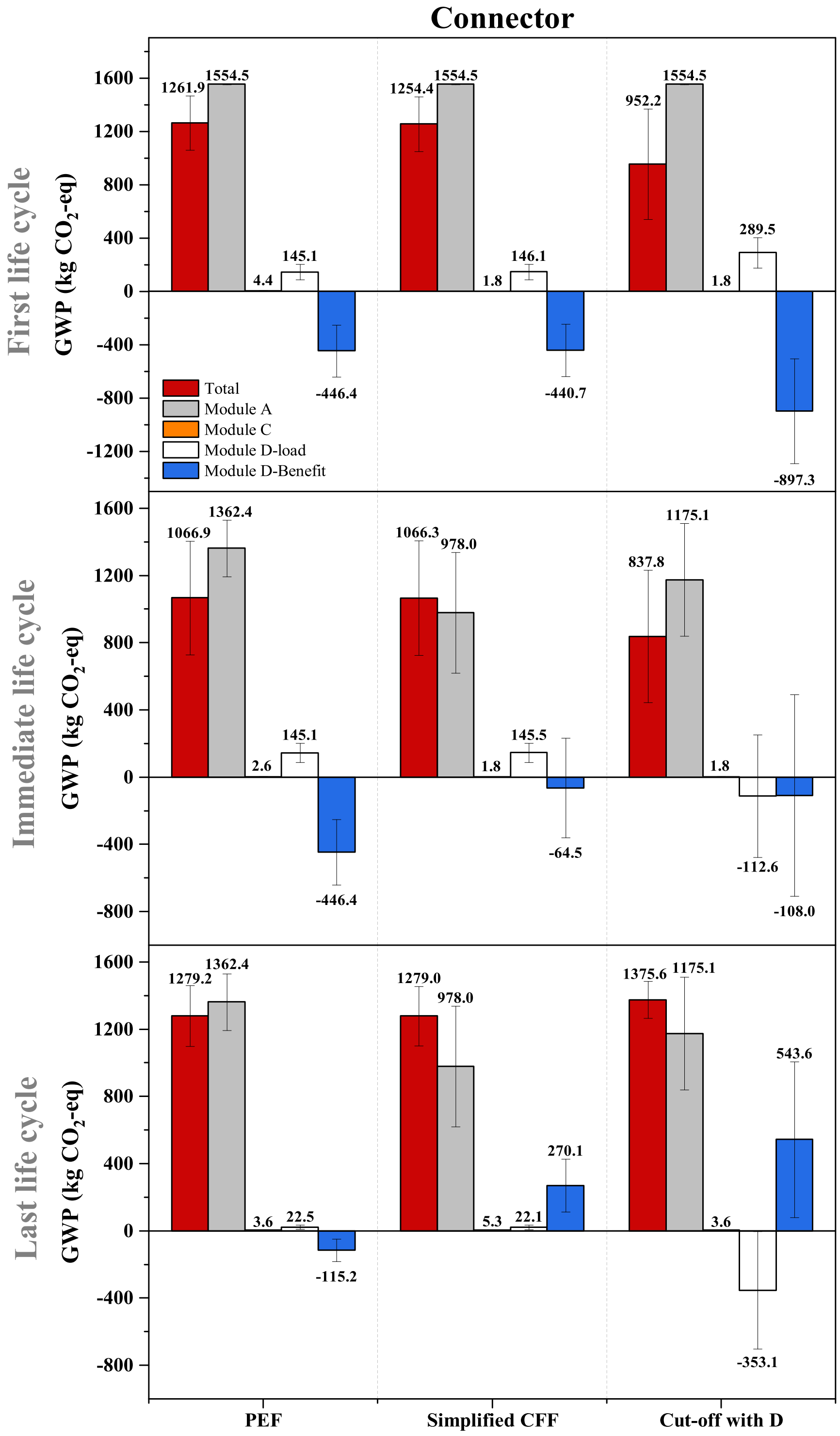

Fig. 6. (*continued*).

Table 5
Sensitivity factors for all impact categories.

	Steel frame	Concrete slab	Steel connector
GWP	R_d, R_c	R_2	R_2
FPMF	R_d	R_2	R_2
OFHH	R_d	R_2	R_2
OFTE	R_d	R_2	R_2
TA	R_d	R_2	R_2
TE	R_d	R_2	R_2

environmental outcomes. However, altering the recyclability rate of the steel frame can have a notable effect on the GWP results, although it is not highly sensitive to other impact categories. In view of this, promoting the recycling of steel frames can be an effective strategy for mitigating GWP.

5. Discussion

Implementing (direct) reuse is expected to yield environmental benefits by extending the building's lifespan, ultimately preventing premature demolition. It is widely recognized that refurbishment (Hasik et al., 2019; Weiler et al., 2017), selective demolition (Akbarnezhad et al., 2014), or upgrading existing buildings (Bragadin et al., 2023) result in more significant environmental benefits compared to demolition and reconstruction. However, these comparisons often lack a holistic approach to understanding the environmental performance of reusing building components across multiple life/use cycles. As a result, the environmental credits and loads in future (re)use cycles remain overlooked.

From an LCA perspective, assessing the potential reuse of modular components represents challenges due to the unknown future value retention processes. The reuse, recycling and disposal of modular components will take place later, and uncertainties surround variables such as the reusability rate, recyclability rate, and amount of waste disposed. To address this concern, a global sensitivity analysis has been conducted in this study to calculate probabilistic life cycle outcomes and identify the key influential factors that significantly affect the LCA results. Therefore, overinterpreting the present results in deterministic or absolute terms should be avoided. This study provides a novel approach to addressing the uncertainties associated with the multiple-use cycles of modular components, an area that has received little attention in the existing body of knowledge.

Previous studies have primarily focused on conducting life cycle assessment of reusing whole buildings (Minunno et al., 2020; Aye et al., 2012), neglecting an in-depth examination of building components, elements, and materials. It is crucial to carry out element- and material-level analyses as different building elements and materials are associated with distinct value retention activities and end-of-use-cycle treatments, which have varying environmental impacts. Identifying the key influential factors that affect the LCA results for individual modular elements allows practitioners to implement appropriate design and deconstruction planning strategies to achieve the desirable reusability level of modular components. For instance, in addition to adopting DfD principles, it is essential to implement proper maintenance and deconstruction procedures to prevent significant damage or minor scratches to the steel frame during the operation and disassembly stages. This strategy will ensure a high level of direct reusability of the core modular component, thereby maximizing the environmental benefits of reuse.

This study presents a novel finding regarding the influence of value retention activities on the environmental consequences of reusing the core modular component and subcomponents. It is found that the environmental impact of the core modular component is more sensitive to repair activities, while that of subcomponents is more sensitive to replacements. During the intermediate use cycle, the environmental impact of reusing the steel frame primarily stemmed from the repair process. Although the steel frame is fully reusable in preceding cycles, repair activities are often necessary to maintain the good condition of the structure after a period of regular use. These repair processes can have a notable influence on the environmental outcomes of steel frame reuse, although the magnitude of the impact is relatively small. It is worth noting that the present analysis only considers surface coating as part of the repair process for the steel frame. Future technological advancements may introduce new value retention practices (e.g., remanufacturing), yielding different environmental impacts (Haupt and Hellweg, 2019). Further exploration is warranted to explore the potential impacts of these value retention processes. On the other hand, the replacement of subcomponents involves the disposal of the replaced parts and the consumption of new virgin materials, resulting in impacts related to landfills and production. This is why the environmental impacts of subcomponents are particularly sensitive to the replacement process. Notably, repairing subcomponents is considered environmentally tolerable as its variation does not have a significant influence on the change in the environmental outcomes.

It is worth noting that the environmental credits associated with concrete recycling are lower than its burdens, whereas the environmental benefits arising from steel recycling are greater than its loads (Fig. 6). Similar findings can be found in previous studies showing that recycling steel could contribute to environmental savings (Abouhamad and Abu-Hamd, 2021). The processes of concrete recycling might induce higher environmental loads (44 ± 29.5 kg co2-eq) than benefits (−3.5 ± 2.0 kg co2-eq), resulting in a positive net environmental impact. The findings echoed prior research reporting the environmental loads of concrete recycling (Abouhamad and Abu-Hamd, 2021). Steel recycling demonstrated clear advantages over concrete recycling, producing remarkable environmental credits (Kröhnert et al., 2022). Therefore, steel structures are recommended for temporary modular buildings, considering their environmental sustainability, especially in multiple-use cycles. Concerning CE considerations for waste treatment at the end of the last use cycle, it has been reported that over 90 % of steel (Sansom and Avery, 2014) but less than 10 % of concrete (Huang et al., 2018) could be recycled. Concrete recycling has long been recognized as a downcycling process, where the emissions and energy demand associated with material recovery may outweigh the benefit of conserving raw resources. However, it is essential to note that the actual impact of concrete recycling still holds potential for improvement with future technological advancements in material recovery (Zhang et al., 2023; Xing et al., 2022).

This study offers a fresh perspective on the environmental viability of different value retention processes for various individual modular components. While the environmental impact of the core modular component is more sensitive to repair and recycling activities, the impact of subcomponents is more sensitive to replacements but not repairs or recycling. These findings suggest that different levels of reusability and recyclability are warranted for core and non-core modular elements. Specifically, it is desirable to achieve a higher level of reusability (i.e., direct reuse) for the core steel frame to avoid the impact of repairs. The finding also suggests that promoting the recycling of the steel frame at its end-of-life can have a significant influence on GWP reductions. On the other hand, a relatively lower level of reusability (i.e., repairable) and recyclability for subcomponents is considered environmentally acceptable. However, the lowest level of reusability, which involves replacement of subcomponents, should be avoided to mitigate the impact of disposal of replaced materials and production of new materials. In view of this, it is crucial to consider appropriate design methods, disassembly procedures, and end-of-use-cycle treatments for individual modular elements and materials, with a view to enhancing their environmental performance of reuse and recycling. In other words, tailor-made design and deconstruction strategies should be developed for different modular components to maximize their reusability and recycling potential while minimizing the multi-use cycle environmental impacts.

6. Conclusions

This study presents the environmental credits and loads of reusing modular components that were allocated across multiple use cycles using three dedicated impact allocation methods, namely, the PEF, CFF, and cut-off with Module D. By comparing the probabilistic LCA results of three dedicated impact allocation methods, the findings suggest that that the PEF approach is more appropriate for interpreting the environmental loads and credits associated with reusing building components over multiple use cycles. This comparison provides a useful guidance for LCA practitioners in selecting an appropriate allocation method specifically tailored to building reuse scenarios.

This study underscores the importance of incorporating value retention processes into the life cycle assessment of building reuses, with a view to enhancing the rigor of the assessment outcomes. Value retention processes, such as repair and replacement, are the necessary activities undertaken to maintain or enhance the value of the modular component and enable continued reuse. By assessing the impacts of the value retention processes, this study offers valuable insights into the potential trade-offs between the environmental benefits and loads of reusing and recycling modular components. These findings can guide structural designers and contractors to implement appropriate design and deconstruction planning strategies, with the goal of avoiding value retention activities that have considerable environmental impacts.

It is important to note that this study has focused on evaluating the environmental impact of reusing three typical modular elements. Similar methodologies can be applied in future studies to assess the multi-life cycle impact of other elements, such as fire boards. Additionally, while this study considered direct reuse, repair, and replacement as value retention processes to extend the lifespan of modular components, the impact of other processes, such as remanufacturing, may require further evaluation. The sensitivity analysis revealed that transport distances or the locations of end-of-use-cycle treatment facilities did not significantly influence the LCA results. This result may be attributed to the assumption that these facilities were situated within or close to Hong Kong territories. In addition to considering the environmental loads and benefits associated with reuse and recycling, their economic and social outcomes should not be neglected. A comprehensive analysis would ideally encompass the environmental, economic and social aspects, providing a holistic life cycle sustainability assessment of circular economy initiatives in the construction sector.

CRediT authorship contribution statement

YY – Conceptualization, Methodology, Formal analysis, Software, Writing – original draft, Writing – review & editing. **BZ** – Formal analysis, Data curation, Software, Writing – original draft, Visualization, Validation. **CL** – Funding acquisition, Data curation, Project administration, Resources, Writing – review & editing. **KY** – Data curation, Investigation, Validation, Writing – review & editing. **AC** – Supervision, Funding acquisition, Writing – review & editing.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

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